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Review Article

Case studies from Germany and Scotland indicate climate change mitigation potential of riverine and coastal wetlands

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Abstract

Wetlands help to regulate the climate by sequestering and storing carbon from the atmosphere into their biomass and soils. Although wetlands can provide valuable ecosystem services such as carbon sequestration, there is a lack of quantifiable data for different types of wetlands. Teal-green carbon in floodplain forests and blue carbon in coastal and marine environments are the focus of this study. We use two contrasting wetland habitats and explore carbon sequestration and storage mechanisms, current evidence and data gaps, and their potential opportunities to contribute to climate change mitigation. Using a case study of hardwood floodplain forests in Germany, we find that age, structure and hydrological conditions are important factors in carbon sequestration while the evidence shows that sequestration is higher in young forests compared to old ones. Soil carbon stocks are found to be larger in the low elevated active floodplain compared to the higher elevated floodplain highlighting the importance of connectivity between the river and the floodplain. In a second case-study exploring carbon storage in Scottish saltmarshes, similar factors driving carbon storage are seen, including, position in the tidal frame, sediment supply, and sediment type. The policy landscape is used to highlight opportunities to increase the potential contribution of Scottish saltmarshes for climate change mitigation. Although data are limited and gaps exist for other greenhouse gas fluxes, overall, we argue that it is crucial to conserve all remaining riverine and coastal wetlands for their biodiversity and carbon storage function. Where possible and practical, these valuable ecosystems should be restored to increase their potential in this regard.

Highlights

- Blue and teal-green carbon habitats can make an important contribution to carbon sequestration and storage.
- Blue and teal-green carbon habitats offer multiple ecosystem services in addition to carbon sequestration.
- Protection and restoration of blue and teal-green carbon habitats provide opportunities for improved climate change mitigation potential.

Key words: Blue carbon, carbon sequestration, climate mitigation, riparian habitats, saltmarsh, teal-green carbon

Introduction to carbon sequestration and storage of floodplain and coastal wetlands

Wetlands are the interface between terrestrial and aquatic environments, which can comprise freshwater or transitional water systems (Kingsford et al. 2016). They support high levels of biodiversity and provide critical ecosystem services yet are threatened globally by habitat loss through conversion or degradation (Kingsford et al. 2016, see Kaden et al. 2026). In this paper, we focus on the climate change mitigation potential that wetland systems can provide, looking at two distinct systems in a riverine and coastal context.

Wetlands are well known for their contribution to water management (e. g. Hughes 1997) however they are also particularly effective at storing organic carbon (OC) for several reasons (Nahlik and Fennessy 2016; Li et al. 2023). Wetlands are often highly productive ecosystems with abundant plant growth. Plants absorb carbon dioxide (CO_2) from the atmosphere through photosynthesis, and when they die, a share of their biomass is transferred into underlying soil, known as 'autochthonous' carbon. This autochthonous supply of OC can be supplemented by 'allochthonous' OC, which is produced outside the system and transported in along watercourses (Ward et al. 2017). If this organic material is successfully trapped and buried, allochthonous carbon can represent a significant additional source of OC to an ecosystem (Smeaton et al. 2024). Wetlands typically have water-saturated soils that create anaerobic, low-oxygen, conditions. On one hand, they can be a source of methane (e. g. see Xiao et al. 2024), but on the other hand, these conditions help to slow down the decomposition of organic matter and can prevent the complete breakdown of plant material, resulting in the accumulation of partially decayed organic matter. This combination of factors leads to high OC accumulation rates and long-term burial in wetlands, when the soils are undisturbed. Wetlands support a variety of plant and microbial life, some of which contribute to OC accumulation through their biological processes. Wetlands often have complex nutrient cycling processes (see Zak et al. 2026), such as nitrogen and phosphorus, which can enhance plant growth and, consequently, carbon capture (Zhang et al. 2022). Water plays an essential role in connecting and transferring organic material and nutrients laterally between habitats along the aquatic continuum from land to sea (Fig. 1) (Ward et al. 2017; Dean et al. 2025).

The scientific community has assigned colours to different forms of organic carbon based on function, attribute, or location, and these terms are continuously adapted and refined as understanding improves (Zinke 2020). Terrestrial carbon, for instance, in forests is called 'green' carbon, carbon stored in the sediments and water of freshwater systems is referred to as 'teal' carbon, while the carbon stored in the coastal and marine environment is known as 'blue' carbon. All of these 'coloured carbon' systems have similar mechanisms for sequestering and storing carbon, dependent on photosynthetic activity to initiate the assimilation of atmospheric CO_2 into biomass, followed by biological transfers of energy and matter into root and soil/sediment for longer-term stor-

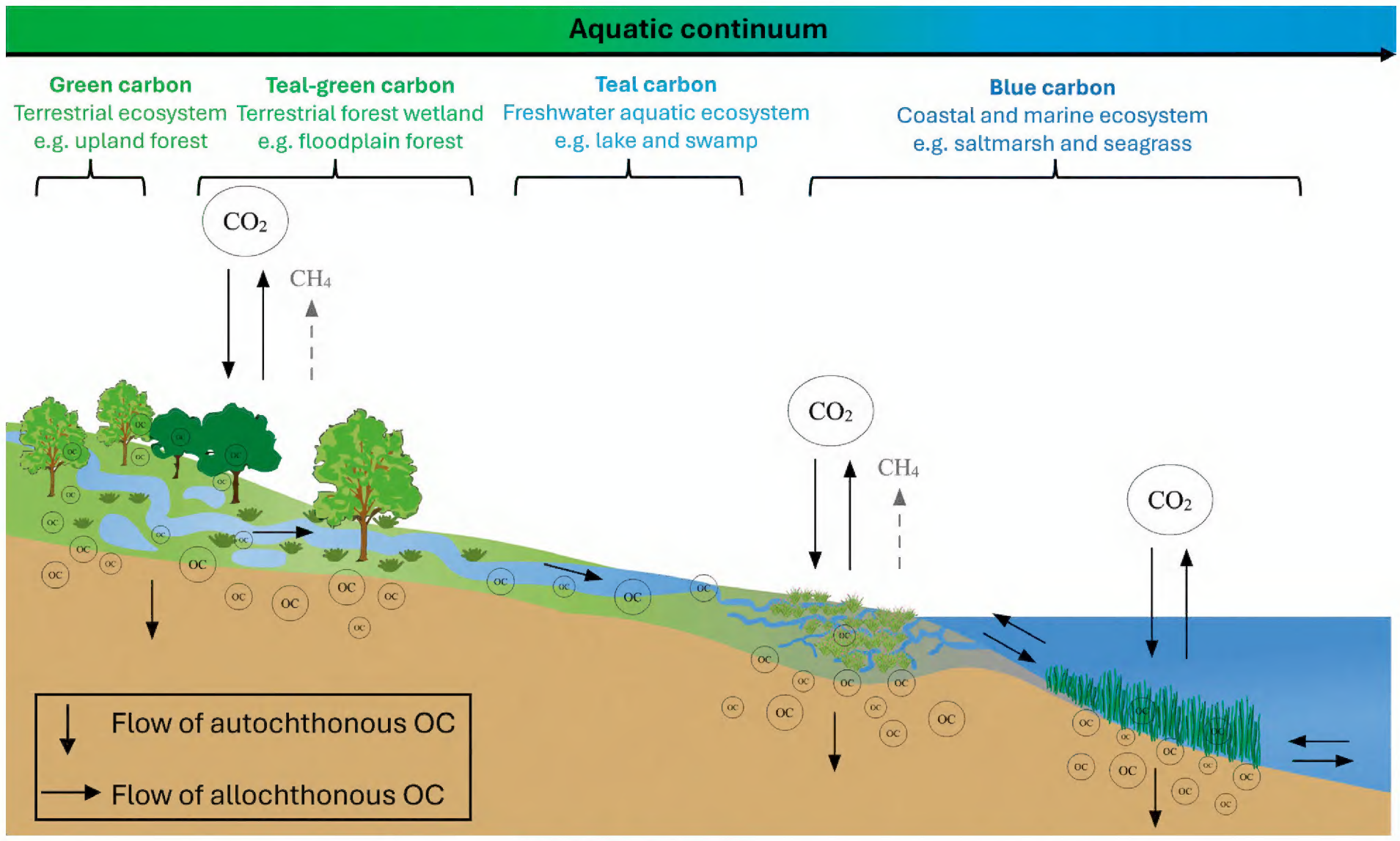

Figure 1. Simplified diagram depicting the capture, transfer, and storage of organic carbon (OC) in freshwater and transitional wetlands along the aquatic continuum from land to sea. The fluxes of CO_2 via photosynthesis and respiration are shown. Wetland habitats of different symbolic colours of carbon sinks have been loosely identified based on their position along the aquatic continuum. Carbon can be stored as organic material within the habitat's biota (short-term) or within the underground root and soil/sediment systems (long-term). Autochthonous carbon is that which is produced locally within each habitat where a proportion can be buried *in-situ*. Allochthonous carbon enters a habitat from an external source, normally via lateral transfer by water. Some of this carbon can also be trapped and buried within the receiving habitat or continue to be transferred elsewhere on its journey to the sea. Methane (CH_4) fluxes are not well known within these systems, signified by the dashed arrow.

age (McLeod et al. 2011; Malerba et al. 2022). For this study, we focus primarily on interface areas, specifically floodplains and salt marshes, to highlight the important function of these wetland systems as natural carbon stores.

Teal-green carbon systems

Teal carbon is a recent term for the OC stored in freshwater ecosystems, including lakes, swamps and rivers (Nahlik and Fennessy 2016; Malerba et al. 2022). Dissolved organic carbon (DOC) and particulate organic carbon (POC) is derived from sources like terrestrial plant material, algae, and other organic matter within the aquatic environment (Carpenter et al. 2011; Williams et al. 2016). Additionally, teal carbon encompasses the carbon stored in sediments at the bottom of freshwater bodies, such as rivers, lakes or ponds, which can accumulate over time and contribute to the overall carbon budget of the ecosystem (Carpenter et al. 2011; Williams et al. 2016).

Green carbon refers to the OC stored by terrestrial ecosystems, including forests and other land-based vegetation such as grasslands, and their soils. It is called green, because plants take up CO_2 from the atmosphere through the process of photosynthesis, which is dependent on the green chlorophyll pigment (Mackey et al. 2008). Green carbon is stored and cycles between pools:

it becomes living biomass via photosynthesis, is partly released again to the atmosphere via respiration when organisms feed on the biomass, and is partly stored in the soil after decay. The residence time of a unit of carbon in each pool varies—the longest being within woody biomass and soil (Roxburgh et al. 2006).

The dynamics of 'teal' and 'green' carbon play a crucial role in the biogeochemical cycling of carbon within wetland ecosystems, as both components are interlinked, influencing processes such as nutrient cycling, ecosystem metabolism, and the storage of carbon. As in other wetlands, the storage of carbon is enhanced within floodplain soils due to waterlogging conditions, which slows decomposition and enhances carbon storage (Heger et al. 2021). Additionally, carbon sequestration of oaks (*Quercus robur*) on the floodplain can be higher than of oaks disconnected from flooding (Shupe et al. 2022b), which can be partly explained by the productivity-enhancing qualities of flood pulses (Junk et al. 1989). Although floodplain forests have previously been defined as 'green' carbon ecosystems, because they are forests (Kiss et al. 2015), their classification as inland freshwater wetlands also matches the definition of 'teal' carbon. Thus, floodplain forests qualify as 'teal-green' carbon ecosystems, as they may store and sequester more carbon than purely 'teal' or 'green' carbon ecosystems. Understanding the dynamics of 'teal' and 'green' carbon is essential for comprehensively assessing the role of terrestrial and freshwater ecosystems in the global carbon cycle and their potential to mitigate climate change.

Blue carbon systems

Blue carbon (BC) refers to the carbon captured through photosynthesis and stored within the coastal and marine environment. The term originated to recognise the disproportionate contribution that vegetated coastal wetland habitats make to carbon sequestration relative to their extent, specifically, mangroves, saltmarshes, and seagrass (Duarte et al. 2005; Nellemann et al. 2009; Serrano et al. 2019). They are understood to be carbon sinks and arguably have substantial potential as a natural climate solution (Macreadie et al. 2021).

Blue carbon ecosystems are found worldwide, typically fringing low-energy intertidal and subtidal coastal zones. Initial studies were focused on tropical systems with high productivity and burial rates (Duarte et al. 2005; McLeod et al. 2011), prompting an expansion of research into these ecosystems globally (Burrows et al. 2014; Macreadie et al. 2019; Krause-Jensen et al. 2022). This has highlighted variability in intra-habitat productivity and between bioclimatic zones (Lavery et al. 2013; Jankowska et al. 2016; Mason et al. 2022). Due to the low oxygen concentrations found in water-logged soils from regular tidal inundation, long-term burial of OC is promoted in these systems, where it may be sequestered for centuries and beyond (McLeod et al. 2011; Macreadie et al. 2019).

Despite increasing numbers of studies for BC systems, there still remain large uncertainties in sequestration rates, and global OC stock estimates vary significantly (Howard et al. 2023). Uncertainties come from several factors, including limited data for habitat spatial extent, highly variable intra and inter-habitat sequestration rates, and poorly quantified fluxes of greenhouse gases beyond CO_2, for instance methane, within coastal wetlands (Macreadie et al. 2021; Eyre et al. 2023). Further research is required to refine the net carbon sink status of these habitats.

Due to the growing interest in nature-based solutions to support climate change mitigation and adaptation efforts, there has been discussion around expanding the definition of BC to include other marine habitats, including but not limited to, kelp, marine sediments, phytoplankton, and calcifying ecosystems (Lovelock and Duarte 2019). Although the coastal and marine environment is also a significant store of inorganic carbon (Smeaton et al. 2021a; James et al. 2024), this fraction is not considered relevant to climate change mitigation policies due to the calcification pathway by which it is formed, a net source of CO_2, and the long timescales over which this carbon accumulates (Fodrie et al. 2017; Macreadie et al. 2017; Turrell et al. 2023). The International Panel on Climate Change (IPCC) AR6 glossary (https://apps.ipcc.ch/glossary/searchlatest.php) currently defines BC as 'biologically driven carbon fluxes and storage in marine systems that are amenable to management', while also recognising that there is potential for the BC concept to be applied to other coastal and marine ecosystems, as clearer evidence emerges. BC ecosystems can be classified as "actionable", "emerging", or "non-actionable", depending on the potential for any management to significantly influence the concentration of atmospheric greenhouse gases (Howard et al. 2023). Positive management activities can include protection and restoration to enhance the carbon sink potential (see de la Vega-Leinert et al. 2026) or, conversely, avoiding human impacts that can result in habitat degradation and associated carbon emissions. Actions to protect and restore BC habitats will increase their potential for climate change mitigation in addition to providing other ecosystem services (Hilmi et al. 2021).

Carbon sequestration in riverine systems: teal-green carbon

Floodplain forests are among the most diverse and productive ecosystems in Europe (see Fig. 2 for an impression). They are extremely valuable given the variety of services they provide (Tockner and Stanford 2002). Floodplain forests provide habitat to unique species of flora and fauna and are seasonal migration corridors

Figure 2. A typical but rare hardwood floodplain forest at the Middle Elbe River, Germany (photo by Dieter Damschen).

for threatened bird species (Hale 2004). They have great potential for OC storage in biomass and soils (Dybala et al. 2019; Heger et al. 2021; Shupe et al. 2022b). Furthermore, floodplain forest soils control nutrient and water availability and they provide runoff regulation (Hughes 1997). These ecosystems are endangered throughout Europe, because of anthropogenic pressures, such as river training and dike construction. Thus, their protection and restoration has increasingly become a topic of public interest (Koenzen et al. 2021, see Birk et al. 2026).

Complex interactions between ground and surface waters, as well as climatic factors such as temperature, evaporative demand, and precipitation, dictate the hydrological dynamics in the floodplain (Krause et al. 2007). The expected increase in frequency and intensity of droughts in Central Europe (Bednar-Friedl et al. 2022), will affect the hydrological balance of the floodplain. Under conditions of water scarcity, the soil matrix plays a pivotal role in controlling rates of transpiration, carbon assimilation, and biomass production by influencing rates of infiltration and depth of percolation (Porporato et al. 2004). In addition, periods of low precipitation and high temperatures are described as the leading factor of forest dieback generally (Allen et al. 2010) and for floodplain forests (Stojanović et al. 2015). Other documented effects of global warming on forests are the reduction of net primary productivity, shifts in community composition, and lengthening of the vegetation period (Porporato et al. 2004; Chmielewski 2007). The many projected effects of climate change on floodplain forest ecosystems widens the gap in our knowledge about the climate mitigation potential of floodplain forests in the future.

Case-study: hardwood floodplain forests at the middle elbe river in germany

Characterising floodplain forests at the Elbe River

The Middle Elbe River is a lowland river with a mean discharge of about 680 m^3 s^{-1} and runs through Germany. In former times, it was a meandering lowland river with abundant floodplain forests. Since the Middle Ages, the construction of dikes and river engineering has led to an 80% loss of the active floodplain (Koenzen et al. 2021). The disconnection of the active floodplain from the river hydrology, through diking, had direct implications on the soil nutrient availability and soil water dynamics, which sustain the floodplain forests (Vásconez Navas et al. 2023). Moreover, the once connected forests have mostly been converted into grasslands in the active floodplain and to arable land in the former floodplain area. Floodplain forests have therefore become scarce, and only 1% of the former forest area exists today (Scholz et al. 2012). To understand the role of these forests in carbon cycling, Shupe et al. (2021) and Heger et al. (2021) quantified the carbon pools of vegetation and soil of the remnant and planted floodplain forests along the Middle Elbe River. Additionally, Shupe et al. (2022a, 2022b) and Shupe (2022) analysed the carbon sequestration rate of dominant *Quercus robur* trees in the same study region.

Methods

Shupe et al. (2021) and Heger et al. (2021) quantified the carbon stocks of hardwood floodplain forests under different hydrologic conditions: in the active

floodplain, in the seepage water zone behind the dike, and in the floodplain of tributaries of the Elbe River. Furthermore, hardwood floodplain forests of varying age (young plantations vs. old forests) and structure (dense and sparse canopy) in the active floodplain were studied. Shupe et al. (2022b) and Shupe (2022) additionally estimated carbon sequestration rates for the dominant *Quercus robur* trees and the overall vegetation, respectively.

To estimate the carbon stock of trees (the main pool of vegetative carbon) first the diameter at breast height (DBH) and the height of the trees was measured in a field campaign. The DBH and the height of the trees were multiplied to calculate the tree volume. From the tree volume, the carbon stock was calculated with species-specific equations. These equations were also used for calculating bark and trunk (BT) aboveground biomass, and biomass expansion factors were used for calculating the root biomass (Shupe et al. 2021). Deadwood, litter, and shrubs were also measured. Dendrochronological methods were applied to tree cores collected from dominant *Quercus robur* trees to estimate forest ages and average growth rates (AGR) for quantifying the carbon sequestration rates of the dominant trees (Shupe et al. 2022a, 2022b). The average carbon sequestration rate of the vegetation in HF forests was calculated by dividing the sum of the trees, shrubs, litter, and deadwood carbon stocks by the estimated forest age.

Soil organic carbon (SOC) stocks were determined to a depth of 1 m (Heger et al. 2021). In order to calculate the SOC stocks, bulk density was measured in collected undisturbed samples (100 cm^3) from 18 soil pits horizon-wise (N = 93 different soil horizons). Moreover, pedotransfer functions (De Vos et al. 2005; Steinicke et al. 2016) were used based on referenced data to calculate the bulk density of remaining horizons (N = 755). Afterwards, SOC stocks were calculated (in Mg ha^{-1}) using the equation defined by Hiederer and Köchy (2011) and Kobal et al. (2011), which includes the SOC in connection to horizon thickness, bulk density, and a correction factor for stoniness which accounts for mineral components > 2 mm.

Carbon stocks in the floodplain forests and sequestration of the vegetation

Average carbon stocks in the vegetation on the active floodplain ranged from 50 Mg ha^{-1} for young plantations to 180 Mg ha^{-1} for old forests (Shupe et al. 2021). The carbon storage of the old forests in the other hydrological conditions (seepage water zone and at tributaries), lay in the same range as the old forests on the active floodplain. Forest SOC stocks ranged between 100 and 150 Mg ha^{-1} (Heger et al. 2021). An active connection to the river had the strongest effect on SOC stocks, with former floodplain sites storing 33% less SOC than the active sites. Within the active floodplains, low sites stored 50% more SOC than high sites (Heger et al. 2021).

For forests of different developmental stages on the low active floodplain, Shupe (2022) estimated a significantly ($p \leq 0.05$) higher average carbon sequestration rate (CSR) of 217 g OC m^{-2} yr^{-1} ± 86 SD for young plantations, than for the old HF forests (see Table 1). The CSR of old dense forests differing in hydrological conditions do not significantly differ ($p \geq 0.05$).

The carbon sequestration rate of the oak trees was higher during flood years in all hydrological conditions, but was significantly reduced by drought only on the active floodplain with a comparatively high elevation

Table 1. The estimated relative organic carbon (OC) sequestration contributions calculated for different types of Hardwood floodplain (HF) forests (based on Shupe 2022).

Forest age and structure	Sequestration Rate (g OC m^{-2} yr^{-1})	Comment on hydrological conditions
Young HF forests	217 ± 86 SD	Young forests on low active floodplain
Old dense HF forests	103 ± 20 SD	Forests on the low active floodplain
Old sparse HF forests	118 ± 22 SD	Forests on the low active floodplain
Old dense HF forests	130 ± 39 SD	Forests on the high active floodplain
Old dense forests	121 ± 36 SD	Forests on the seepage water zone
Old dense forests	116 ± 35 SD	Forests at tributaries

(Shupe et al. 2022b). Dominant *Quercus robur* trees on the Middle Elbe floodplains had an annual AGR of 2.2 mm $year^{-1}$ for the low active floodplain, the seepage water zone and the tributary, while the AGR in the high active floodplain was higher with 2.8 mm $year^{-1}$ (Shupe et al. 2022b).

Discussion and evidence gaps

Carbon storage in hardwood floodplain trees can range between 8 to 281 Mg C ha^{-1} (Sutfin et al. 2016). Our mean value of 180 Mg C ha^{-1} lies within this range, however is lower than the estimated 281 Mg C ha^{-1} for hardwood floodplain (HF) forests at the Danube River (Cierjacks et al. 2010). For other forest types outside of floodplains, the average carbon stocks in aboveground and belowground biomass, deadwood, and soils are reported to be 224 Mg ha^{-1} in Germany (Wellbrock et al. 2017). This estimate is 33% less than in our studied HF forests with estimates of 330 Mg ha^{-1} for vegetation and soils combined (in the low elevated active floodplain). Additional biomass expansion factors for branches, fruit, and leaves could have been applied to increase the carbon stock estimates, but were omitted to compensate for the stock losses due to wind damage and hollowed trunks which were commonly observed. This could be a reason for the difference to other values in the literature, where these expansion factors were used. Leaves and nuts were also measured separately. Our estimate is therefore conservative, and the carbon stocks of the floodplain forest vegetation could be higher.

With regard to the sequestration rates, the average range of 103–217 g OC m^{-2} yr^{-1} is within the 15–400 g OC m^{-2} yr^{-1} range estimated for HF forests in Europe (Vilà et al. 2013; Rieger et al. 2017). As is also seen in Table 2 for the broadleaf forests of the UK, the younger forests have larger sequestration rates, which slow down when the forests are older. Nevertheless, values from 103–130 g OC m^{-2} yr^{-1} are still reasonable carbon sequestration rates given the rates for the soils are still missing as these remain a knowledge gap for our studied floodplain forests.

The average growth rates of the dominant *Quercus robur* trees in the Middle Elbe River are comparable to the 2.1 to 3.0 mm $year^{-1}$ AGRs of *Quercus robur* in floodplain forests found in the Czech Republic (Tumajer and Treml 2016) and slightly higher than AGRs of 1.7 to 2.5 mm $year^{-1}$ on hydromorphic soils in north-east Germany (Scharnweber et al. 2013). Within upland forests and parks in north-east Germany, *Quercus robur* have overall lower AGRs of 1.4 to 1.8 mm $year^{-1}$ and 1.1 to 2.0 mm $year^{-1}$ respectively, compared to the floodplain trees (Scharnweber et al. 2011).

Table 2. The estimated relative organic carbon (OC) sequestration contributions from some of Scotland and the UK's terrestrial, coastal and marine habitats. It is important to note that these values come with uncertainties and it can be difficult for direct comparisons because of the variable protocols and standards in data collection between terrestrial and marine systems.

Habitat	Scottish Areal Extent (km^2)	Sequestration Rate (g OC m^{-2} yr^{-1})	Comment	Reference
Saltmarsh	58.4	87.45	Average value for Scottish saltmarshes estimated from mud-rich and sand-rich saltmarsh sites and accounting for the upper and lower marsh zones using multiple chronological markers.	Miller et al. 2023
Seagrass	15.04	67.91	Rate is based on six intertidal seagrass meadows within the Solent, England.	Lima 2020
Fjord Sediments	2,608	57.1 ± 10.9	This value is based on OC burial rates within Scottish fjords. There are very limited data for C sequestration rates for the wider shelf seas; however, rates are likely to be higher within fjords relative to the shelf seas.	Smeaton et al. 2021b
Peatlands	17,000	16–27* -0.27**	Values are based on UK-wide data. Variable rates, dependent on habitat condition. When degraded, peatlands are known to be a carbon source.	* Ratcliffe et al. 2018; Evans et al. 2017; Aitkenhead and Coull 2019 ** Rees et al. 2018
Broadleaved mixed woodlands	1,337	381 (average over 30 years) * 191 (average over 100 years) * 14* / 26–46 ** (soil carbon accumulation)	Values are based on UK-wide data	* Gregg et al. 2021 ** Rees et al. 2018

Soil organic carbon stocks in the floodplain soils of the Elbe River reached values of 150 Mg C ha^{-1}, which are higher than 35 Mg C ha^{-1}, the median from 598 observation in 76 study areas reported by Dybala et al. (2019) in a global synthesis and meta-analysis of riparian forests carbon stocks. Regarding carbon sequestration in floodplain forest soils, knowledge gaps exist for German floodplain forests. In a study comparing the differences of carbon sequestration between depressional wetlands and floodplain soils of Czech and Midwest USA, mean organic C sequestration values of 47 + 14 g C m^{-2} $year^{-1}$ were reported for floodplain soils (Craft et al. 2018). This suggests that the soils of the HF forests of the Middle Elbe River are probably also contributing to carbon sequestration that we have not accounted for in this study. Overall, soil carbon sequestration rates remain uncertain. Long-term impacts of drought, warming, and altered hydrology on OC storage are poorly understood until now.

We conclude that hardwood floodplain forests can store a considerable amount of OC and that floodplains are natural carbon sinks. Restoration of former floodplains, for instance by dike relocation projects, could sequester additional carbon in their soil and vegetation (Heger et al 2021; Shupe et al 2021, 2022b; Vásconez Navas et al. 2023). At the same time, they support the characteristic flora and fauna of floodplains and offer space for flood water retention (see Fink et al. 2026).

Opportunities for floodplains

The importance of conserving floodplain ecosystems has been recognised over the last few decades (Koenzen et al. 2021), and numerous policy initiatives have been introduced to enhance protection and sustainable

management of these valuable habitats in Germany. More recently, the German Federal Nature Conservation Act (BNatSchG) has been revised to strengthen legal protection for floodplain areas, emphasising the importance of maintaining natural flood regimes and habitat connectivity. The EU Habitats Directive also plays a key role, requiring member states to designate protected areas and implement management plans that safeguard floodplain biodiversity. The 'Source-to-Sea' approach underscores the importance of integrated catchment management—recognising the connectivity between upstream water regulation, floodplain habitats, and downstream ecosystems, including coastal and marine ecosystems (e.g., Kellock et al. 2023). Such holistic perspectives are vital for reducing flood risks, sustaining biodiversity, and maintaining ecosystem services such as water purification, carbon sequestration, and habitat provision. Policies are increasingly emphasising the importance of restoring dynamic floodplain processes to enhance resilience against future climate impacts. Under the EU Nature Restoration Law (EU 2024/1991), Germany is committed to achieving significant progress in restoring destroyed habitats by 2030. The regulation sets targets for different habitats: by 2030, at least 20 percent of land and marine area in the EU are to be restored. Rivers and floodplains are among the habitats prioritised for restoration, for example by removing river barriers (Darre et al. 2025).

Funding mechanisms are also being expanded to prioritise floodplain conservation and restoration. For example, the national Blue Belt program (German: Bundesprogramm Blaues Band) of the German Government aims at restoring natural hydrological processes in floodplain regions across Germany (Borges et al. 2016). This national program is complemented by regional projects that focus on reconnecting rivers with their floodplains, restoring wetlands, and enhancing ecological corridors. The ethos is to create resilient river landscapes that are valuable in terms of natural environment and, at the same time, offer attractive places for human wellbeing.

Policy challenges remain in balancing economic development, agriculture, and flood protection objectives with ecological needs. Drainage and channelization practices, historically widespread to support agriculture, continue to pose threats to the natural function of floodplains. Efforts have to be made to phase out such interventions and promote nature-based solutions that utilise natural processes to manage flood risks sustainably. Stakeholder engagement must become a cornerstone of policy implementation, with regional partnerships promoting collaboration among government agencies, landowners, NGOs, and research institutions. This would facilitate knowledge exchange and support projects focused on habitat restoration, flood management, and climate resilience. Climate change introduces additional uncertainty and urgency, with increased flood events and altered hydrological cycles necessitating adaptive management strategies. Germany's floodplain habitats hold immense ecological and societal value yet face significant policy and management challenges. Continued efforts to integrate legal protection, innovative restoration projects, stakeholder collaboration, and climate adaptation measures are essential to ensure that these vital ecosystems are preserved and resilient for the future.

Climate change mitigation and adaptation mechanisms of Blue Carbon habitats

Blue Carbon habitats are well adapted to dynamic coastal environments, inundated to different degrees by daily tides, and, in good condition, can function effectively as long-term organic carbon (OC) stores. Besides capturing atmospheric CO_2 via photosynthesis, their physical vegetative structure enables a high trapping capacity for sediments transported into the system (Fagherazzi et al. 2012). This includes allochthonous OC, originating from both terrestrial and marine systems (Smeaton et al. 2024). Saturated soils also protect BC stores from risks such as fire, to which terrestrial carbon stores can be vulnerable.

Sediment trapping, a process which enables OC accumulation and burial, is a critical mechanism for the longer-term survival of these coastal habitats against relative sea level rise (Fagherazzi et al. 2012, see Ibáñez et al. 2026). As sediment is trapped, these habitats can accrete vertically to keep pace with rising sea levels highlighting an adaptation to climate change impacts. In situations where sediment supply is deficient, sea-level rise can outpace vertical accumulation, leading to habitat loss through erosion or drowning (Kirwan et al. 2010; Krauss et al. 2014). However, recent research suggests that there may be a critical threshold rate of sea level rise, above which, coastal habitats such as saltmarsh may struggle to survive due to natural feedback mechanisms resulting in auto-compaction of sediment (Saintilan et al. 2022). Another adaptation to rising sea levels is the ability of blue carbon ecosystems to expand laterally into space through the colonisation of sediment deposits by vegetation (Saintilan et al. 2022). Often, however, the availability of accommodation space on the coast is limited due to human development or natural embankments, resulting in a physical constraint to this potential expansion. This lack of available space compromises the habitat's potential to adapt naturally to rising sea levels and could result in significant global losses in the future (Schuerch et al. 2018).

In addition to their OC storage service, BC habitats also support specialist biodiversity, and contribute to coastal protection and coastal community well-being, livelihoods, and food security (Barbier et al. 2011; Himes-Cornell et al. 2018). Despite the significant value attributed to them (Barbier et al. 2011; Luisetti et al. 2019), these habitats are threatened and declining globally due to coastal development, unsustainable extraction, pollution, and climate change impacts, including sea level rise and increased storminess (McLeod et al. 2011; Schuerch et al. 2018; Macreadie et al. 2021; Neiske et al. 2025). Disturbance to these habitats can lead to the rapid decomposition of OC stored in soils over long timescales through the introduction of oxygen (Coverdale et al. 2014; Lovelock et al. 2017). Loss of these habitats results in direct greenhouse gas emissions and indirectly, a loss of future carbon sequestration potential (Pendleton et al. 2012). Given the timescales involved in accumulating large underground stores of carbon, protection of existing BC habitats should be considered as a priority to prevent further avoidable emissions (Smeaton et al. 2024). Furthermore, the restoration and enhancement of BC habitats can reverse the loss of habitat and habitat condition, and help tackle the joint biodiversity and climate crises (Hilmi et al. 2021; Macreadie et al. 2021; Jones et al. 2024).

Case-study: saltmarshes of Scotland

Characterising Scottish saltmarshes

Scotland has a wealth of BC habitats within its extensive coastal and marine environment (Burrows et al. 2014; Smeaton et al. 2021a; Cunningham and Hunt 2023). These habitats all contribute significantly to Scotland's biodiversity; however, it is primarily the coastal vegetated systems, saltmarshes and sea-grass beds, that fall within the category of 'actionable' BC systems (Howard et al. 2023). Scotland also has vast stores of carbon within its seabed sediments, particularly in the fjord systems of the west coast, where organic-rich material accumulates in glacially, over-deepened basins (Smeaton et al 2017; Smeaton et al. 2021b). Seaweeds, including kelp, are highly productive ecosystems and extensive in Scotland's waters, however their lack of root systems, limits their contribution to longer-term carbon stores. Both seabed sediments and sea-weeds are considered to be 'emerging' blue carbon systems in Scotland, with a view that further evidence will help to better understand the climate mitigation potential of these habitats (Cunningham and Hunt 2023).

A national saltmarsh survey for Scotland estimated approximately 240 saltmarshes amounting to an area of 58 km^2 distributed around the coastline (Haynes 2016), and representing 13% of the total UK saltmarsh habitat (Austin et al. 2021). The Scottish coastline is highly variable, and saltmarsh has de-veloped in diverse geomorphological settings (see Fig. 3 as an example). The largest saltmarshes, greater than 1 km^2, cover the low-lying 'firths' or estuaries of the eastern and south-west coast, while smaller pockets of habitat exist on the west coast within glacially formed landforms, such as at the head of sea lochs, or perched on cliff tops (Haynes 2016).

Saltmarshes are valued habitats and over 70% of Scotland's saltmarshes are notified features within national and internationally designated protected areas. However, despite the high levels of protection, Scottish saltmarsh extent has declined over time (Beaumont et al. 2014). Historical land claim over the last 400 years for industrial development and agriculture is estimated to have taken approximately 50% of the mudflat and saltmarsh within the Forth estuary (Han-som et al. 2001; Beaumont et al. 2014). Today, pressures range from coastal development, climate change impacts including relative sea level rise, pollution, historical embankments, and poor grazing management (Haynes 2016).

Saltmarsh blue carbon

In recognition of the important role that saltmarshes play globally in OC seques-tration and storage, several recent studies have sought to quantify the service provided by Scottish saltmarshes (Austin et al. 2021; Smeaton et al. 2022; Mill-er et al. 2023; Smeaton et al. 2024; Deary 2025). These studies have highlighted saltmarshes as one of Scotland's most significant BC habitats contributing to coastal OC storage (representing ~ 50% of the coastal OC stock) and as having the highest OC sequestration rate of any coastal or marine habitats (Cunning-ham and Hunt 2023). An estimated 0.37 ± 0.09 Mt OC (to a standardised 0.1 m sediment depth) is held in Scottish saltmarshes and approximately 4,385 ± 481 tonnes of OC are added annually; 85% of this OC originates from terrestrial sources (Smeaton et al. 2022, 2024; Miller et al. 2023).

Figure 3. Thrift (*Armeria maritima*) flowering on the saltmarsh at high tide on the Isle of Harris, Western Isles, Scotland (©Lorne Gill/NatureScot).

However, the use of average values for OC stocks and rates, over-simplifies what is a complex system and can lead to misleading messaging, highlighting the importance of obtaining representative data for national-scale audits. Saltmarshes exist around the world and the interplay between physical processes, climate, geology, and human pressures shapes the ecological structure and function of the ecosystem differently (Yando et al. 2023). Scottish saltmarshes are a good example of how OC storage and sequestration vary between and within sites as a function of i) biomass density (high marsh zones hold more OC in biomass and soils than low-mid marsh zones), ii) sediment composition (sandy sediments hold less OC than muddy sediments), iii) tidal frame (lower zone observed to have higher sequestration rates attributed to more frequent tidal inundation) and iv) geomorphology (accreting systems will capture more sediment) (Austin et al. 2021; Miller et al. 2023; Smeaton et al. 2023; Deary 2025).

Within the context of Great Britain, Scotland's saltmarshes store an average 6.31 ± 1.56 kg OC m^{-2} in their surficial sediments; this is ~23% higher than the average OC stock normalised for area of saltmarshes in Great Britain, which is attributed to the higher inputs of allochthonous OC from the adjacent organic-rich land, and because they are older and have therefore been accumulating for longer than those found in the south of the UK (Smeaton et al. 2022; Smeaton et al. 2024). At a global scale, however, Scottish saltmarshes have a sequestration rate below the global average, a consequence of its temperate climate and higher proportion of sandy soils (Miller et al. 2023).

Evidence gaps for Scotland's saltmarshes

Despite the number of studies investigating the OC storage service of Scotland's saltmarshes, several evidence gaps remain. The national saltmarsh survey for Scotland was undertaken over 10 years' ago (Haynes 2016); without a contemporary survey for comparison, it is impossible to understand the rate of change in extent and condition. Traditional survey techniques are resource and time intensive; thus, alternative innovative methods for surveying are being investigated using earth observations data and modelling approaches (Fitton et al. 2021; Archer-Rand et al. 2022).

While there is a good understanding of the amount of OC sequestered by saltmarshes, other greenhouse gas fluxes, including methane, are still relatively unknown. Methane emissions are understood to be suppressed by saline inundation (Kroeger et al. 2017), therefore emissions may be expected in the upper saltmarsh typically receiving a higher freshwater contribution, compared to the pioneer and lower saltmarsh zones. Additionally, the role of carbonate burial in BC budgets needs to be further investigated (Saderne et al. 2019; Van Dam et al. 2021). It is also unclear what impact typical management practices, such as grazing and drainage, and human pressures that impact habitat condition, have on the OC sequestration capacity of saltmarshes (Graversen et al. 2022). Finally, climate change impacts, for instance changes to precipitation or storm frequencies, might affect OC sequestration (Wang et al. 2021). Filling these gaps is essential to ensure that saltmarshes can be optimised to realise their climate mitigation service.

Opportunities for saltmarshes

Since Scotland declared a climate emergency in 2019, stressing the need for urgent action, several opportunities towards the increased protection and enhancement of the natural environment, including for BC habitats, have emerged. Scottish Government has recently launched its Scottish Blue Carbon Action Plan, outlining how key evidence gaps will be filled to better integrate blue carbon into future policy and on-the-ground protection. Other recent policies, such as the Scottish Biodiversity Strategy and Scottish National Adaptation Plan, will strengthen the consideration of the natural environment within decision-making. Concepts of 'Source to Sea' (Kellock et al 2023) and seascape connectivity (Preston et al 2025) are increasingly referenced, with opportunities to become incorporated into emerging policies that take a holistic approach to environmental management and protection by recognising connectivity between catchment ecosystems. For coastal wetlands, this could help to manage pressures upstream and downstream, to enhance natural function and support ecosystem services that include carbon sequestration (Kellock et al 2023; Preston et al 2025). Establishment of entities such as the Scottish Blue Carbon Forum and the UK's Blue Carbon Evidence Partnership (UKBCEP) has improved the engagement of stakeholders and enhanced the collective understanding of how BC resources contribute to climate change adaptation, mitigation, and resilience, through a coordinated approach to research and improvement of the evidence base. The UKBCEP's Evidence Needs Statement identifies the evidence needed to support the UK in advancing its BC commitments (UKBCEP 2023). This includes working towards the potential inclusion of saltmarsh within the UK's Greenhouse Gas Inventory. In addition, to underpin policy commitments

from Scottish Government, several funds for nature restoration have been established, such as the Nature Restoration Fund and the Scottish Marine Environment Enhancement Fund, supporting investment in restoring the natural environment to tackle the joint biodiversity and climate crises. In addition, funding through the Facility for Investment Ready Nature in Scotland has supported exploration of new mechanisms for investing in natural capital. Several projects supporting saltmarsh enhancement have been initiated through these funds, ranging from the development of a domestic carbon code to physical restoration (e.g. Carter et al. 2024). Enhancement and restoration of Scottish saltmarshes could add to the potential of these habitats to play a role in climate change mitigation, however protecting existing saltmarshes to avoid emissions from disturbance to existing long-term carbon stores should arguably be the priority (Smeaton et al 2024). The combination of policy, research and collaboration, and funding to build capacity and communicate the wider benefits of nature, offers significant opportunities to support the role of saltmarsh as a Nature-based Solution for climate mitigation and adaptation.

Overall conclusions

We provide two examples of contrasting wetland habitats in a European context to highlight their potential contribution to climate change mitigation. Current evidence for these wetland systems, including floodplains and coastal saltmarshes, indicates they are among the most efficient ecosystems for carbon sequestration per unit area in Europe. Although they cover a relatively small portion of the Earth's surface, their OC sequestration potential relative to other habitats is significant. However, as their spatial extent is limited, they cannot be regarded as a 'silver bullet' to climate change, but should be considered as being part of the toolbox of nature-based climate change mitigation strategies. Further studies and monitoring are required to fill evidence gaps (e. g. the role of methane) and improve our understanding of carbon cycling dynamics within and between wetland ecosystems. Disturbance to these systems from anthropogenic activities can result in degraded habitat condition and atmospheric emissions, while the accelerating impacts of climate change increase uncertainty about how much OC these ecosystems can continue to sequester and store. Protecting the existing carbon stocks of floodplains and coastal saltmarshes is therefore critical, while further carbon sequestration could be promoted by reduction of anthropogenic pressures or restoration—not only for climate change mitigation, but also for the many ecological and adaptation benefits of these ecosystems, such as flood protection, habitat for plants and wildlife, and water purification. Government policy can provide a significant mechanism under which these beneficial actions can be realised (see Klusmann et al. 2026).

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Conflict of interest

The authors have declared that no competing interests exist.

Ethical statement

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Use of AI

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Author contributions

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Data availability

All of the data that support the findings of this study are available in the main text.

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